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CLIMATE
CHANGE
RESEARCH
REPORT
CCRR-32



*Responding to
Climate Change
Through Partnership*

An Assessment of the Vulnerability of Forest Vegetation of Ontario's Clay Belt (Ecodistrict 3E-1) to Climate Change



Sustainability in a Changing Climate: An Overview of MNR's Climate Change Strategy (2011-2014)

Climate change will affect all MNR programs and the natural resources for which it has responsibility. This strategy confirms MNR's commitment to the Ontario government's climate change initiatives such as the Go Green Action Plan on Climate Change and outlines research and management program priorities for the 2011-2014 period.

Theme 1: Understand Climate Change

MNR will gather, manage, and share information and knowledge about how ecosystem composition, structure and function – and the people who live and work in them – will be affected by a changing climate. Strategies:

- Communicate internally and externally to build awareness of the known and potential impacts of climate change and mitigation and adaptation options available to Ontarians.
- Monitor and assess ecosystem and resource conditions to manage for climate change in collaboration with other agencies and organizations.
- Undertake and support research designed to improve understanding of climate change, including improved temperature and precipitation projections, ecosystem vulnerability assessments, and improved models of the carbon budget and ecosystem processes in the managed forest, the settled landscapes of southern Ontario, and the forests and wetlands of the Far North.
- Transfer science and understanding to decision-makers to enhance comprehensive planning and management in a rapidly changing climate.

Theme 2: Mitigate Climate Change

MNR will reduce greenhouse gas emissions in support of Ontario's greenhouse gas emission reduction goals. Strategies:

- Continue to reduce emissions from MNR operations through vehicle fleet renewal, converting to other high fuel efficiency/low-emissions equipment, demonstrating leadership in energy-efficient facility development, promoting green building materials and fostering a green organizational culture.

- Facilitate the development of renewable energy by collaborating with other Ministries to promote the value of Ontario's resources as potential green energy sources, making Crown land available for renewable energy development, and working with proponents to ensure that renewable energy developments are consistent with approval requirements and that other Ministry priorities are considered.
- Provide leadership and support to resource users and industries to reduce carbon emissions and increase carbon storage by undertaking afforestation, protecting natural heritage areas, exploring opportunities for forest carbon management to increase carbon uptake, and promoting the increased use of wood products over energy-intensive, non-renewable alternatives.
- Help resource users and partners participate in a carbon offset market, by working with our partners to ensure that a robust trading system is in place based on rules established in Ontario (and potentially in other jurisdictions), continuing to examine the mitigation potential of forest carbon management in Ontario, and participating in the development of protocols and policies for forest and land-based carbon offset credits.

Theme 3: Help Ontarians Adapt

MNR will provide advice and tools and techniques to help Ontarians adapt to climate change. Strategies include:

- Maintain and enhance emergency management capability to protect life and property during extreme events such as flooding, drought, blowdown and wildfire.
- Use scenarios and vulnerability analyses to develop and employ adaptive solutions to known and emerging issues.
- Encourage and support industries, resource users and communities to adapt, by helping to develop understanding and capabilities of partners to adapt their practices and resource use in a changing climate.
- Evaluate and adjust policies and legislation to respond to climate change challenges.

An Assessment of the Vulnerability of Forest Vegetation of Ontario's Clay Belt (Ecodistrict 3E-1) to Climate Change

William C. Parker, Stephen J. Colombo, and Mahadev Sharma

Ontario Ministry of Natural Resources, Ontario Forest Research Institute,
1235 Queen St. E., Sault Ste. Marie, ON P6A 2E5

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Applied Research and Development
Ontario Forest Research Institute
Ministry of Natural Resources
1235 Queen Street East
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Canada P6A 2E5

Telephone: (705) 946-2981
Fax: (705) 946-2030
E-mail: information.ofri@ontario.ca

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Summary

The vulnerability of the forest vegetation of Ontario's northern Clay Belt region to climate change was assessed using forest tree species composition and forest productivity as indicators. Changes in species composition were examined using the modelled bioclimatic niche of 15 tree species under current and future climate projected for three periods (2011-2040, 2041-2070, 2071-2100) using four general circulation models and two emissions scenarios (A2, B1). Using climate projections from an ensemble model for these same scenarios and periods as inputs to climate-based site index and genecological models, changes in height growth were examined for several major tree species of the region as a measure of effects of climate on forest productivity. Major northward geographic shifts in species bioclimatic niche were projected, resulting in suitable climatic habitat decreasing for boreal forest species of the region, and becoming more favourable for species currently associated with more southern, i.e., Great Lakes-St. Lawrence, areas. In general, results from the site index and genecological models suggest that where soil moisture availability remains relatively unchanged, initial warming over the next few decades will improve the growth of several boreal species. Collectively, these results suggest that assisted migration of seed sources and other climate change adaptation strategies directed specifically at maintaining productivity within the Clay Belt may not be necessary until mid-century.

Résumé

Évaluation de la vulnérabilité de la végétation forestière de la ceinture d'argile de l'Ontario (Écodistrict 3E-1) aux changements climatiques

On a évalué la vulnérabilité de la végétation forestière de la ceinture d'argile du Nord de l'Ontario aux changements climatiques en utilisant les essences d'arbres et la productivité de la forêt comme indicateurs. On a examiné les changements de la composition des essences à partir du créneau bioclimatique modelé de 15 essences d'arbres en fonction du climat actuel et du climat projeté pour trois périodes (2011-2040, 2041-2070, 2071-2100) et au moyen de quatre modèles de circulation généraux et de deux scénarios d'émissions (A2, B1). À partir des projections climatiques d'un modèle d'ensemble pour les scénarios et les périodes comme données de base d'un indice de site et de modèles génécologiques, on a examiné les changements de la croissance des hauteurs de plusieurs essences principales de la région comme mesure des effets du climat sur la productivité de la forêt. On a projeté des mouvements géographiques vers le Nord des créneaux bioclimatiques des essences, donnant lieu à la diminution de l'habitat climatique convenable des essences de la forêt boréale de la région et à la plus forte prévalence d'un climat favorable aux essences actuellement associées à des zones plus au sud, comme la région des Grands Lacs et du Saint-Laurent. En général, les résultats de l'indice du site et des modèles génécologiques portent à croire là où l'humidité des sols demeure relativement la même, le réchauffement des quelques prochaines décennies améliorera la croissance de plusieurs essences boréales. Dans l'ensemble, ces résultats donnent à penser que la migration provoquée de graines et le recours à d'autres stratégies d'adaptation aux changements climatiques visant expressément à maintenir la productivité de la ceinture d'argile pourraient ne pas être nécessaires avant le milieu du siècle.

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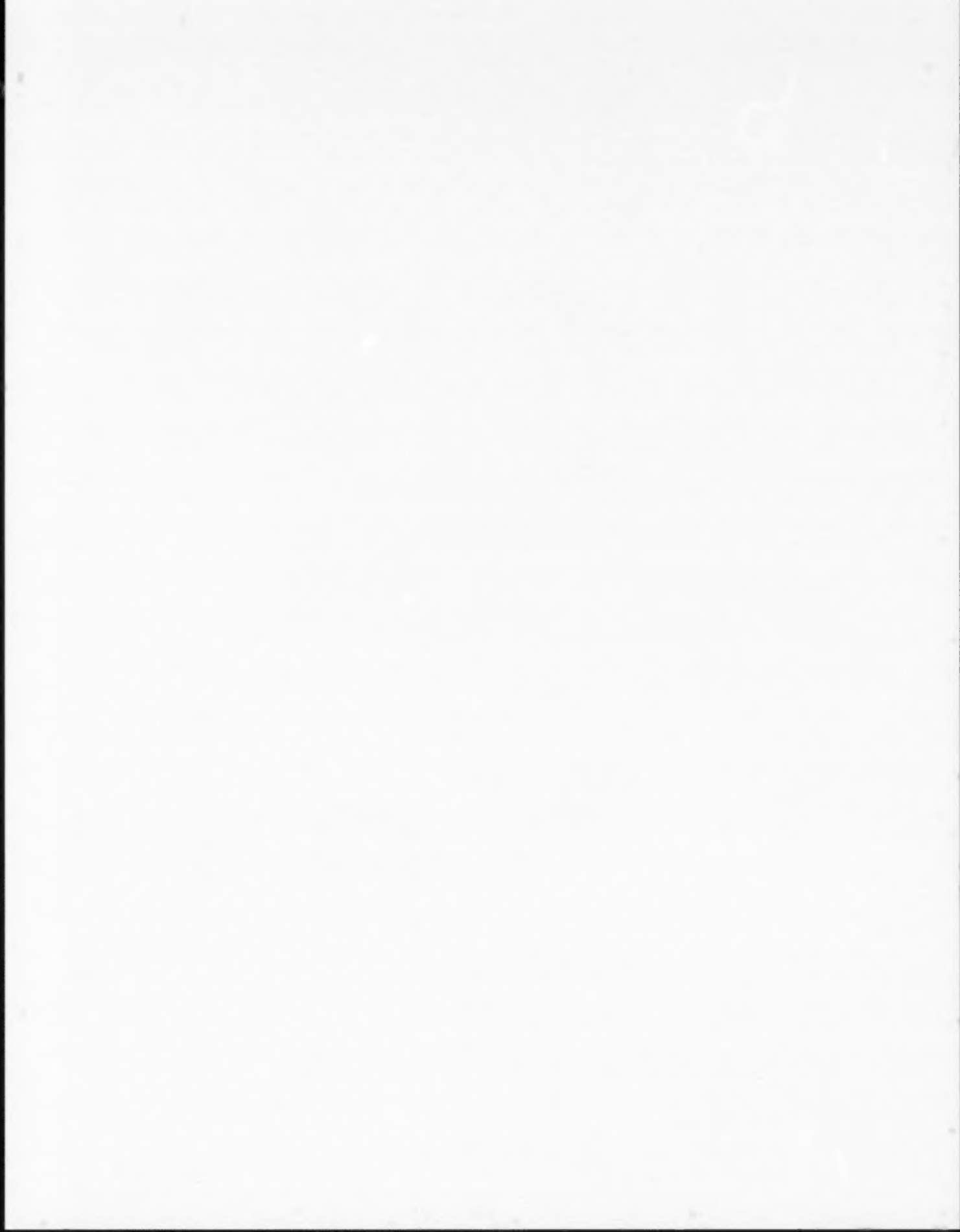
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Foreword

This is one in a series of reports to help resource managers evaluate the vulnerability of natural assets to climate change. Given that vulnerability assessment techniques continue to evolve, it is important for resource managers to learn by doing and to pass on knowledge gained to support MNR and others engaged in adaptive management. Accordingly, the vulnerability assessment reports included in the Climate Change Research Report Series have been prepared using the best available information under the circumstances (e.g., time, financial support, and data availability). Collectively, these assessments can inform decisionmaking, enhance scientific understanding of how natural assets respond to climate change, and help resource management organizations establish research and monitoring needs and priorities.

Cameron Mack

Acting Director, Applied Research and Development Branch



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Introduction

Continued human-generated atmospheric emissions of carbon dioxide (CO₂) and other greenhouse gases (GHG) are projected to increase global average temperature by as much as 6.4 °C by the end of the century (IPCC 2007). A warmer, more variable climate is predicted to be accompanied by regionally altered precipitation regimes and increased frequency of extreme weather (IPCC 2007). Greater warming and increased precipitation are expected in Canada than in more southerly locations of North America. For Ontario, increases in mean annual temperature of 3 to 6 °C and annual mean precipitation of 5 to 7% are projected by the end of the century (McKenney et al. 2010).

Predicted ecological effects of climate change on Ontario's forests include increased disturbance by fire and pests, reduced productivity where soil moisture becomes limiting, and changes in species composition (Parker et al. 2000, Lempière et al. 2008, Williamson et al. 2009). Likewise, climate change will affect the forest sector through the cost, quantity, quality, and timing of access to timber through operational or salvage harvest, with potentially significant adverse effects on northern, forest-based communities (Browne and Hunt 2007, Johnston et al. 2009, Williamson et al. 2009). Climate change may reduce future timber supply in Ontario by 2 to 4% by the 2050s, with an annual economic cost of between CDN\$1 and \$7 billion (NRTEE 2011).

Forest adaptation to climate change can be defined as adjustments in ecological, social, or economic systems to moderate vulnerability and enhance recovery of forest ecosystems (Smit et al. 2000). Given that climate has already changed and many projected effects are already occurring (Soja et al. 2007, Lempière et al. 2008), adaptation is essential even if sustained reduction in global GHG emissions is achieved. Adapting forests to climate change will require unprecedented effort to minimize damage and to exploit opportunities that may arise (Lempière et al. 2008, Johnston et al. 2009, Williamson et al. 2009). Key to adaptation decisionmaking are regional vulnerability assessments to understand current and future exposure and sensitivity to climatic stress, recognizing the many uncertainties associated with climate change (Lempière et al. 2008, Johnston et al. 2009, Williamson et al. 2009). When applied within an adaptive management framework, vulnerability assessments can facilitate timely, cost-effective adaptation strategies (Johnston et al. 2009, Williamson et al. 2009). Moreover, the costs of well-planned adaptation strategies may be far less than those involved in reactive responses to effects (Lempière et al. 2008, NRTEE 2011).

Vulnerability assessments are increasingly being used to guide resource managers in deploying adaptation strategies to respond to climate change (Williamson et al. 2009, Gleeson et al. 2011, Swanston et al. 2011). In this context, vulnerability is defined as "the degree to which a system is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes" (IPCC 2007). Vulnerability is a function of the exposure, sensitivity, and adaptive capacity of a system, with exposure and sensitivity positively related and adaptive capacity negatively related to vulnerability. The Ontario Ministry of Natural Resources (OMNR) has committed to completing vulnerability assessments as a component of regional climate change adaptation efforts (Gleeson et al. 2011, OMNR 2011), using a participatory approach and a framework that embraces adaptive management (Figure 1). In this report, we present the results of a preliminary climate change vulnerability assessment for terrestrial forest vegetation within the Northeast Clay Belt region of Ontario (Ecodistrict 3E-1), with forest species composition and forest productivity as indicators of vulnerability. An overview and synthesis of the results of the complete vulnerability assessment conducted for the Clay Belt are reported by Lalonde et al. (2012).



Figure 1. Steps in a vulnerability assessment to inform development of climate change adaptation strategies (Gleeson et al. 2011).

Background

Climate and physiography

Clay Belt Ecodistrict 3E-1 covers roughly 4.13 million ha of Ecoregion 3E (Lake Abitibi Ecoregion) in northeastern Ontario (Figure 2). The area has a humid mid-boreal ecoclimate with long, cold, and snowy winters and short, warm summers (Crins et al. 2009). The ecodistrict is located on the Precambrian Shield with largely granitic and gneissic bedrock, with organic soils, luvisols, and gleysols being common (Baldwin et al. 2000). Deep glaciolacustrine sediments (28%) and morainal calcareous clays and silts (38%) are prominent features. The Clay Belt is characterized by poor to imperfect drainage, with a relatively large (26%) area of organic deposits. Coniferous forest occupies about one-third (35%) of the area. Black spruce (*Picea mariana* (Mill.) BSP) is the dominant conifer species and is widely distributed across both lowland and upland sites, growing with lesser amounts of white spruce (*Picea glauca* (Moench) Voss), white cedar (*Thuja occidentalis* L.), larch (*Larix laricina* (Du Roi) K. Koch), and balsam fir (*Abies balsamea* L.). About one-quarter (28%) of the ecodistrict is covered by mixedwood forests of white birch (*Betula papyrifera* Marsh.), trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.), black and white spruce, and balsam fir. The drier, better drained glaciofluvial sands occupying about 6% of Ecodistrict 3E-1 are dominated by jack pine (*Pinus banksiana* Lamb.), with a small (6%) area of aspen- and birch-dominated hardwood forests occupying well-drained sandy sediments along rivers and lakes. Treed bogs and other wetland vegetation types are limited to about 5% of the area, largely associated with lowland black spruce forest. The economies of the larger communities within the ecodistrict depend primarily on forestry, mining, tourism, and trapping. Very little of the regional landbase is committed to commercial agricultural production, with small areas of oats, barley, hay, and potatoes, and some beef cattle and other animals (OMAFRA 2012).



Figure 2. Ecoregion map of Ontario showing location of Ecodistrict 3E-1.

Pre-settlement disturbance regimes

Wildfire is the predominant stand-replacing natural disturbance agent in Canada's boreal forests, and shaped the structure, function, and composition of the pre-settlement forest in the ecodistrict (Kneeshaw and Gauthier 2003). Prior to European settlement in the late 1800s, the mean fire interval of this region varied from about 80 to 420 years (Bergeron et al. 2010), with lowland conifer forests having a longer fire interval than upland conifer and mixed forests (Bergeron et al. 2001). Where the fire cycle is relatively long (>125 yr), insects, disease, and high winds become important natural disturbance agents (Harper et al. 2003, Kneeshaw and Gauthier 2003). The primary insect species causing periodic, broad-scale disturbances are eastern spruce budworm (*Choristoneura fumiferana* Clemens), jack pine budworm (*Choristoneura pinus pinus* Freeman), and forest tent caterpillar (*Malacosoma disstria* Hübner) (Fleming et al. 2000, deGroot et al. 2005). Diseases typically cause finer-scale disturbances, and include root rot (*Armillaria* sp. and *Inonotus tomentosus* (Fr.:Fr.) Teng) and white pine blister rust (*Cronartium ribicola* J.C. Fischer) (Fleming et al. 2000, deGroot et al. 2005). Catastrophic stand-replacing disturbance of boreal forests by severe wind damage due to thunderstorms, tornadoes, and cyclonic winds is much less frequent than fire (i.e., 1,000 to 2,000 year return intervals) but can affect large areas (≈10,000 to 30,000 ha) (Newmark 1984, Flannigan et al. 1989, Peterson 2000). Many forests in Ecodistrict 3E-1 occupy sites with shallow or poorly drained soils that limit rooting depth and are vulnerable to frequent (≈10 yr), localized windthrow that occurs at wind speeds ≥70 km hr⁻¹ (Gardiner 1975, Smith et al. 1987).

Anthropogenic disturbance of Ecodistrict 3E-1 was minimal prior to European settlement. Small populations of nomadic native peoples inhabited this area as early as 4900 yr BP, and subsisted through hunting, fishing, and gathering of edible plant materials. These peoples had negligible effects on the structure and composition of forests. Prior to 1850, European settler activity in northern Ontario was confined to the fur trade and the operation of scattered trading posts with minimal local effects on forests. Agricultural settlement began as early as 1877, but remained at a small subsistence scale until the end of the century (Randall 1940).

Post-settlement, human influences on the forest

Near the turn of the century, increased interest in minerals, timber, and new areas of arable land led to the survey, exploration, and settlement of a large, uninhabited region called "New Ontario" that contained Ecodistrict 3E-1 (Kent 1966, Lambert 1967, Pugh 1975). Authors of an 1899 report on the geology and natural resources of the Clay Belt suggested that much of the area was ideal for agricultural development. From 1903 to 1931, settlement was aggressively promoted, with highest population densities aggregating along railroads constructed early in the century (Randall 1940, McDermott 1961). During peak settlement efforts, an estimated 806,000 ha of the Clay Belt were cleared of forests and farmed (Pugh 1975). In the first few decades of the 20th century, severe fires also destroyed large areas of forest in Ecodistrict 3E-1 due to a combination of abnormally dry summer weather, railroad construction, and burning for land clearing (Lefort et al. 2003). Most notably, a fire centred on the towns of Porcupine and Cochrane burned more than 2,200 km² in 1911, and the Matheson fire of 1916 destroyed an additional 2,600 km² of forest. Despite provincial incentives, by the 1930s the short growing season, summer frosts, poorly drained soils, and economic factors discouraged all but the heartiest of farming settlers, with most abandoning their land and the region (Randall 1940, McDermott 1961).

Logging and the establishment of small, local sawmills to access the boreal forest of the region also began at the start of the 20th century. Commercial harvesting (primarily the selective horse logging of large, quality individual trees) expanded significantly around 1912 with the establishment of Abitibi Power and Paper at Iroquois Falls and Spruce Falls, Co. in Kapuskasing. Clearcutting began to replace selective logging as the preferred silvicultural system in the 1920s (Carleton 2000). Because the extensive areas of poorly drained clay lowlands made forest access difficult, logging was confined to the winter months. From 1930 to 1950, clearcut harvest for pulpwood expanded, with many mills constructed in the region and wood increasingly transported via all weather roads and railroads. Gradual replacement of horse logging by mechanized harvest in the 1950s to 1960s significantly increased both the total area and size of individual cut areas and allowed for summer harvest, but led to problems with rutting, soil compaction, and damage of conifer regeneration in lowland spruce forests (Lefort et al. 2002). Harvest methods designed to reduce site disturbance and protect advance regeneration are now commonly applied in the Clay Belt (Lefort et al. 2002).

From 1951 to 1995, 24% of Ecoregion 3E was harvested, primarily by clearcutting, with some single cuts larger than 5,000 ha (Perera and Baldwin 2000). Little effort was given to active forest regeneration until the 1970s when planting effort was increased (Randall 1940, Carleton 2000). Until the 1990s, clearcutting was pervasive and Ontario's boreal forests were managed almost exclusively for timber production. However, recognition of the importance of biodiversity, ecological processes, and the role of forests in global biogeochemical cycles led to passage of the Crown Forest Sustainability Act (Statutes of Ontario 1995) in 1994, legislating greater emphasis on ecological and social values of forests through the use of "ecosystem management" and practices that emulate natural disturbance patterns at stand to landscape scales (OMNR 2010).

Over the past 100 to 130 years, human utilization of the forest resource has significantly altered the structure and composition of forests in Ecoregion 3E from their pre-industrial condition (Carleton and MacLelland 1994, Carleton 2000, Pinto et al. 2008). In general, the proportion of intolerant hardwood forests has increased and conifer-dominated forests decreased (Hearnden et al. 1992, Carleton 2000, Pinto et al. 2008). At the species level, the frequency and relative abundance of jack pine, white pine (*Pinus strobus* L.), and red pine (*Pinus resinosa* Ait.), larch, and balsam fir is lower, and prevalence of white birch, poplar, white cedar, maple (*Acer* sp.), and spruce is higher than prior to settlement. This has been attributed to a host of human and natural factors, including the preferential harvest but little planned regeneration of conifer species, extensive clearcutting, effective fire suppression, and increasing damage by exotic pathogens such as white pine blister rust and European larch sawfly (*Cephalcia lariciphila* (Wachtl)) (Pinto et al. 2008).

A change in forest condition in the Clay Belt also likely resulted from intentional human alteration of the disturbance regime, from disturbance by wildfire to harvesting of forests largely by clearcutting (Carleton 2000, Perera and Baldwin 2000, Cyr et al. 2009). The stand-replacing fire interval has increased over the past 160 years, likely due to a combination of a cooler, wetter climate and increasingly effective fire suppression (Bergeron et al. 2001, Bridge 2001). In conifer forests growing on lowland clay and organic deposits, the mean fire interval

has increased from an historic value of about 520 years to a current value of >1,920 years (Bergeron et al. 2001). For the period 1921 to 1995, mean fire intervals of 900 to 1,930 years were reported within Ecodistrict 3E-1, with a trend towards longer intervals from west to east (Bridge 2001). Generally, upland conifer forests are characterized by more frequent, higher intensity, stand-replacing fires, while mixedwoods and lowland conifers experience fires of variable intensity and have longer fire intervals (Van Sleetun 2006, Crins et al. 2009). Fire suppression has also greatly reduced the occurrence of low intensity surface fires, with 95% of the annual area burned resulting from a few large fires (Van Sleetun 2006). This has resulted in replacement of fire-dependent, shade intolerant, early successional species by more fire sensitive, shade tolerant, later successional species (Carleton 2000).

Recent change in climate

Over the past century, global mean temperature has increased by 0.74 °C, with the recent rate of warming averaging 0.13 °C per decade (IPCC 2007). This warming was associated with greater temperature increases at more northerly latitudes, significant regional changes in precipitation regime, and increased frequency of high temperature and heavy precipitation events (IPCC 2007, Min et al. 2011, Coumou and Rahmstorf 2012). This change in climate has strongly affected terrestrial ecosystems, altering the timing of phenological events, inducing species range shifts and growth declines, and affecting patterns of forest disturbance by fire and pests (Parmesan 2006, IPCC 2007, Soja et al. 2007). Climate change is now occurring faster than originally projected, suggesting potential rapid, non-linear change rather than a slow linear adjustment in the Earth's climate (Soja et al. 2007). In Ontario, a significant increase in annual daily minimum temperature, decreased diurnal temperature range, and increased annual precipitation occurred from 1950 to 1998 (Zhang et al. 2003). Mean historical values for several climatic variables for Kapuskasing suggest this trend towards warmer temperatures, higher precipitation, and longer growing seasons has also occurred in Ecodistrict 3E-1 (Table 1).

Table 1. Historical, current, and future projected values of several climatic variables for Ecodistrict 3E-1 (i.e., Kapuskasing, Ontario). Future values are ensemble projections of four general circulation models for three periods under the A2 and B1 emissions scenarios.

Climate change scenario	Period	Mean annual Ta* (°C)	Mean GS Ta (°C)	GSL (days)	GDD (°C)	Mean annual Ppt (mm)	GS Ppt (mm)	Annual Pet-Ppt (mm)
A2	2011-2040	2.5	13.3	182	1460	886	528	61
	2041-2070	4.2	14.3	193	1730	913	550	89
	2071-2100	6.2	15.5	205	2064	956	592	119
B1	2011-2040	2.3	13.4	177	1411	867	501	65
	2041-2070	3.3	13.7	187	1565	931	555	64
	2071-2100	4.0	14.0	193	1662	911	548	82
Current	1971-2000	0.76	12.7	165	1198	829	461	72
Historical	1931-1960	0.74	12.6	166	1194	786	426	91
	1900-1930	0.30	12.5	159	1130	674	386	105

* Abbreviations: Air temperature (Ta), growing season (GS), growing season length (GSL), growing degree days ≥ 5 °C (GDD), precipitation (Ppt), and potential evapotranspiration (Pet) calculated using mean monthly Ta (Dunne and Leopold 1978). Growing season begins when mean daily temperature ≥ 5 °C for five consecutive days after March 1, and ends when minimum daily temperature < -2 °C after August 1.

For details, see http://cfs.nrcan.gc.ca/projects/37?lang=en_CA.

Future disturbance regime

Human and natural disturbances shape forest ecosystems by influencing their structure, composition, and functional processes. Climate change is expected to increase the rate and extent of natural disturbances (Dale et al. 2001). A higher occurrence and intensity of disturbance events will affect forests more than changes in mean temperature and precipitation regimes alone since disturbance will initiate reorganization of forest vegetation, and therefore accelerate shifts to species compositions that are better adapted to the future climate (Jentsch et al. 2007, Lloret et al. 2011). Climate change effects on natural disturbance regimes projected for Ecodistrict 3E-1 are

discussed in other vulnerability assessments (see Brown and McAlpine in review, Waller et al. in review), but their relevance to future forest composition and productivity is summarized briefly below.

Climate change is forecast to increase fire frequency and area burned in boreal Canada due to increased lightning and human ignitions, longer fire seasons, and increased fire weather severity (Soja et al. 2007, Lempière et al. 2008). The annual area burned in the larger Clay Belt region of Ontario and Quebec was predicted to increase by more than 60% by 2100, but this falls within the range of natural variability over the last 6,500 years (Bergeron et al. 2010). In Ontario's Ecoregion 3E, the annual number of fires—both natural and human-caused—is projected to increase by 5% by 2040 and 28% by 2090 (Wotton et al. 2005). Correspondingly, estimates indicate that the area burned may increase by up to 25% by 2040 and as much as 25 to 50% by 2090. If fire suppression resources remain at present levels, the number of fires that escape initial attack could increase 30% by 2040 and 80% by 2100 (Wotton et al. 2005). In contrast, minimal net change in the number of fires and a slight decline in overall fire hazard over the next 60 years were estimated for Ecodistrict 3E-1 by Brown and McAlpine (in review). However, they predicted that the length of the fire season could increase by 11 (2041) to 24 (2071) days due to a projected earlier start and later end to this period.

Climate change may disrupt associations between native forest insects and diseases and their hosts as well as their natural enemies, leading to shifts in the timing of insect outbreaks and changes to damage extent and severity (Ayres and Lombardero 2000, Bale et al. 2002, Sturrock et al. 2011, Jactel et al. 2012). Temperature is a primary climatic driver of insect population dynamics, with warming increasing population growth by increasing the number of generations in a season and decreasing overwinter mortality (Ayres and Lombardero 2000, Bale et al. 2002). Similarly, increased damage by some tree diseases is anticipated where temperature, precipitation, and humidity changes favour native pathogens (Anderson et al. 2004, Sturrock et al. 2011). Warmer temperatures may also allow range shifts of insect species to higher latitudes and elevations (Bale et al. 2002). For example, the area defoliated by eastern spruce budworm in Ontario over the next few decades is projected to increase by roughly 25% due to northward extension of the species range in warmer climates (Candau and Fleming 2011). Increased drought, heat, and other abiotic stresses of forests may predispose trees to greater injury and mortality by certain functional groups of insects and pathogens that prefer unhealthy host trees (Allen et al. 2010, Dietze and Moorcroft 2011, Sturrock et al. 2011).

Climate change may increase the prevalence of forest decline and dieback, exemplified by sudden aspen decline recently observed over large areas in the western US and Canada (Hogg et al. 2008, Rehfeldt et al. 2009). Drought events, projected to increase in occurrence, are often an inciting or contributing factor to decline and dieback (Wang et al. 2012). Aspen decline observed in northeastern Ontario and Ecodistrict 3E-1 in the 1990s was associated with combination of repeated forest tent caterpillar defoliation and drought, and was most severe in older stands occupying nutrient poor sites (Candau et al. 2002). Incidence of winter freeze-thaw events, with reduced snow depth, may damage xylem and fine roots, reduce water uptake and transport, and predispose mature white birch and other species to drought and heat stress and secondary attack by pests (Auclair et al. 1997, Hennon et al. 2012).

Increased climate variability and extreme weather are expected to result from human-driven climate warming (Easterling et al. 2000, Salinger 2005, IPCC 2007). Although extreme meteorological events are difficult to predict, future climate of the boreal forest globally will likely be characterized by more hot days and nights, heat waves, heavy precipitation events, and widespread drought (IPCC 2007, Rummukainen 2012). A higher frequency of extreme wind events is predicted for the Clay Belt (Waller et al. in review). Warmer than normal spring temperatures leading to earlier dehardening and budbreak, coupled with increased frequency of late spring frost, could lead to more frequent occurrence of severe frost damage to conifer regeneration as noted in spring 2007, near Kapuskasing, Ontario (Man et al. 2009). Since many biological processes are governed by temperature and precipitation, future increases in extreme weather may be more important ecologically than gradual changes in mean climate (Jentsch et al. 2007, Knapp et al. 2008, Zimmermann et al. 2009).

Methods and general approach

In this assessment, we used four general circulation models (GCM) and two GHG emissions scenarios (A2, B1). The Canadian (CGCM31), Japanese (MIROC3.2MR), Australian (CSIROMK3.5), and United States (NCARCCSM3) GCMs were used to generate current historical (1971-2000) and future projected values of selected climate variables in two GHG emissions scenarios (McKenney et al. 2011).¹ The A2 scenario represents continued population growth and heavy reliance on fossil fuel energy. In the B1 scenario, global population growth is similar to that in A2 but heightened use of resource-efficient technologies reduces emissions below current levels by mid century. For each GCM and emissions scenario, climate values were generated for three, 30-year periods: 2011 to 2040, 2041 to 2070, and 2071 to 2100. Ensemble climate projections for a given scenario and period were estimated as the arithmetic mean of the four GCM projections. For all climate projections, Kapuskasing, Ontario, located near the centre of Ecodistrict 3E-1, was used as the geographic reference point. Historic, current, and future projected values of selected climatic variables that are known to influence plant distribution and productivity are presented in Table 1.

Forest species composition

Bioclimate envelopes (BCE) (i.e., area over which a species' fundamental bioclimatic niche occurs) for individual tree species were generated using thin plate smoothing and ANUCLIM climate envelope software in combination with georeferenced species occurrence data compiled from several sources (for details see McKenzie et al. 2007). Current BCEs for 15 dominant tree species of the boreal and Great Lakes-St. Lawrence (GLSL) forest regions of Ontario were produced using historical (1971-2000) climatic station averages. These BCEs were then used to delineate future area of the species' BCE as projected by the four GCMs for the A2 and B1 scenarios and three periods. The results do not imply that tree species will migrate to keep pace with and occupy the BCE in its future location. The BCE of each species was further subdivided into "core" and "marginal" climatic area. The core climate area is defined by the frequency distribution of tree occurrence bounded by 5% and 95% of total observations, and marginal climate area is represented by infrequent and extreme occurrence observations (Figure 3).

Changes in forest composition were estimated as the proportional area of Ecodistrict 3E-1 containing core climate occupied by a given species historically and in three future periods based on projections from two GCMs (CGCM3.1 and MIROC3.2MR) under the A2 scenario. Temporal changes in species richness associated with shifts in BCE were expressed as the total number of species with core climate within the ecodistrict for a given period. We focused on this small subset of tree species because BCEs for relatively rare species are more uncertain. As well, we report changes in BCE of "core" area as these are likely more indicative of potential species response under climate change. Although the influence of climate change-induced changes in disturbance are not accounted for, we recognize that stand-replacing disturbances (fire, insect outbreaks, harvest, etc.) are an important factor that will determine the presence/abundance of tree species within the current BCE (He et al. 2002, Xu et al. 2012). Finally, we focus our results and discussion on BCEs generated using the A2, high future emissions projection, given that current emissions rates are beyond this level. Examples of BCEs produced for six selected Ontario tree species using CGCM3.1 and the A2 scenario are presented in Appendix 1. For a more complete treatment, readers are directed to the BCEs for 130 North American tree species for several GCMs, scenarios, and time periods that were generated using this same modelling approach (available at <http://planthardiness.gc.ca/index.pl?m=16&lang=en>).

¹ http://cfs.nrcan.gc/projects/3?lang=en_CA

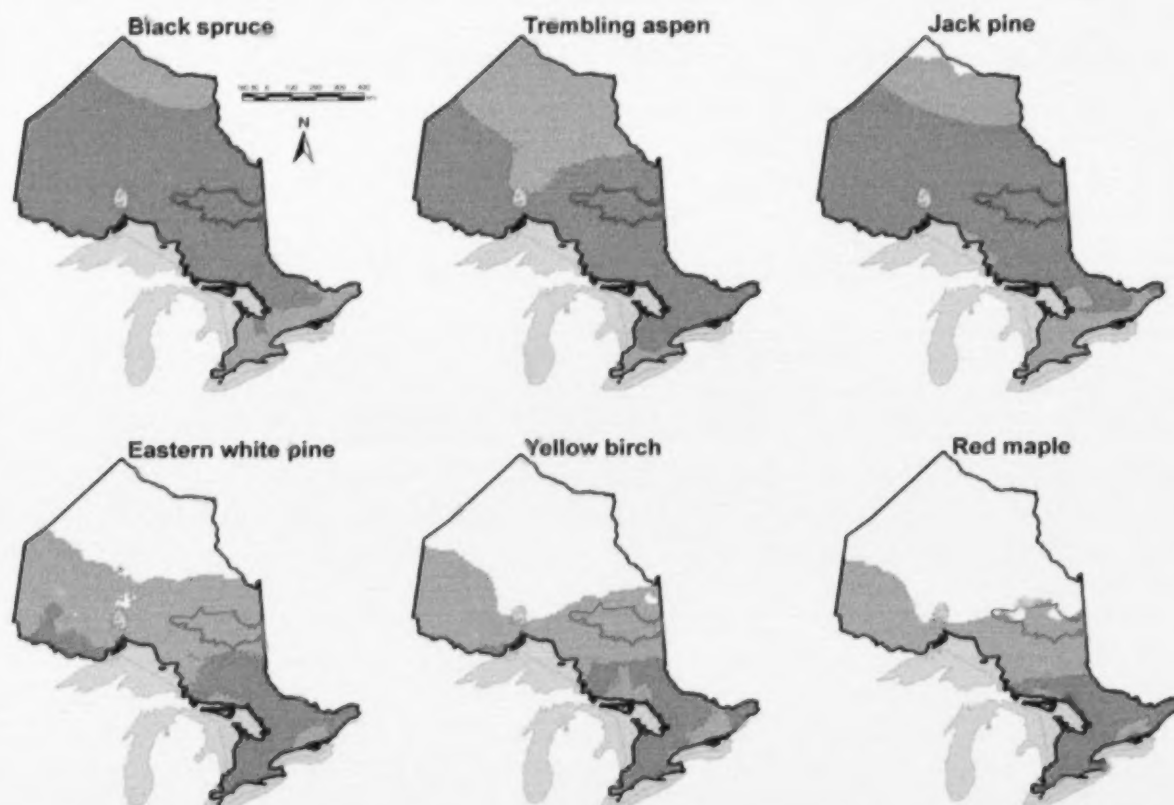


Figure 3. Current bioclimate envelopes for six Ontario tree species. Green represents core climatic range, brown denotes marginal climatic range, and white indicates the climate where the species does not presently occur. The red line encloses the area of Ecodistrict 3E-1.

Forest productivity

Climatic effects on forest productivity (i.e., dominant and co-dominant tree height) for several ecologically and commercially important boreal tree species were estimated using three alternative empirical site index (SI) models obtained from the scientific literature. Site index, defined as the average height of dominant and co-dominant trees (top height) at a specified (index) age, is recognized as an efficient means of expressing productivity. Climate-based SI models derived for plantations (black spruce and jack pine) (Sharma et al. in review) and natural stands (black spruce, jack pine, balsam fir, trembling aspen, and white birch) (Ung et al. 2001, McKenney and Pedlar 2003) were used to estimate total tree height at age 40. These models were selected because they were derived from data from regions geographically relevant to our assessment, i.e., Ontario (McKenney and Pedlar 2003, Sharma et al. in review) and Quebec (Ung et al. 2001).

Climate-based SI models for black spruce and jack pine plantations were derived using empirical data for three plots within each of 25 plantations for each species from across northern Ontario (Sharma et al. in review). Climate variables reflective of annual and growing season temperature and precipitation and the length of the growing season were used as inputs in species-specific models (Table 2). The biophysical SI models of Ung et al. (2001) were derived for balsam fir, black spruce, trembling aspen, and white birch using data collected from more than 6,600 plots in natural stands in the boreal forest region of Quebec. These biophysical SI models are empirical functions of four climatic and one edaphic variable, with each value expressed as a second-order polynomial. Climatic variables were functions of annual growing degree days, growing season precipitation, an annual aridity index, and growing season cumulative vapour pressure deficit (Table 2). For all predictions using this model, we used a soil water holding capacity for sandy loam soils 50 cm in depth with 5% stone content.

Table 2. Site index (a) and genecological (b) models used to predict climate change effects on growth of several boreal tree species. Genecological models are subdivided by transfer functions and population response functions.

Model type/species	Phenotypic trait	Age	Growth model	Reference
a. Climate-based site index models				
Black spruce, jack pine	Height (m)	40	$b1 / (1 - (1 - b1/h35) * (0.875)^{b2})$, where $b1$ and $b2 = f$ (temperature, precipitation regime, growing season length)	Sharma et al. in review ^a
Black spruce, balsam fir, trembling aspen, white birch	Height (m)	40	$b10 + b11 * SI^{b12} * (1 - e^{-b13 * age^x})^x$, where $x = b14 * SI^{b15}$ and $SI = f$ (temperature and precipitation regime, and soil moisture-holding capacity)	Ung et al. 2001 ^a
b. Transfer functions				
White spruce	Height (m)	22	$484.149 + 0.199 (dSP) + 1.459 (dMAXT) - 4.525 (dMAXT)^2$	Andalo et al. 2005 ^a
Jack pine	BAI (m ²)	25	$1.01 + 0.003 (SP) - 0.062 (MAXT)$	Savva et al. 2007 ^b
Jack pine	Height (m)	9	$0.339 (dMINT) - 0.017 (dMINT)^2$	Carter 1996 ^c
White spruce	Height (m)	22	$0.692 (dMINT) - 0.056 (dMINT)^2$	Carter 1996 ^c
Balsam fir	Height (m)	15	$0.943 (dMINT) - 0.115 (dMINT)^2$	Carter 1996 ^c
White pine	Height (m)	16	$0.248 (dMINT) - 0.040 (dMINT)^2$	Carter 1996 ^c
Response functions				
Jack pine	Height (m)	39	$17.8 / 1 + (August MINT - 10.66 / 5.482)^2$	Thomson and Parker 2008 ^a
Black spruce	Height (m)	36	$9.27 / 1 + (February MINT - 17.46 / 11.09)^2$	Thomson et al. 2009 ^a
White spruce	Height (m)	24	$7.42 / 1 + (January MAXT - 7.44 / 4.59)^2$	Cherry and Parker 2003 ^a

Abbreviations: Height at age 35 years (h35), basal area increment (BAI), difference between current and future growing season precipitation (dSP), difference between current and future mean annual maximum temperature (dMAXT), difference between current and future mean annual minimum temperature (dMINT), mean annual maximum temperature (MAXT), mean annual minimum temperature (MINT).

^a Dependent variable is predicted absolute value of the trait.

^b Dependent variable is predicted proportional change relative to the trait value under the current climate.

^c Dependent variable is predicted percentage change relative to the trait value under the current climate.

To estimate current height, the independent variables were assumed to be the mean climate value observed for empirical plot data and the biophysical SI term of the model set equal to 1 (see Table 4, in Ung et al. 2001). Using a regression tree approach, climate and soil-based SI models were derived for natural black spruce and jack pine stands using data collected from 1,140 plots covering the boreal forest region of Ontario (McKenney and Pedlar 2003). These regression models used organic layer and mineral soil depth, and several temperature and precipitation values as independent variables. The regression tree approach provides predictions that are based on categorical (e.g., mean annual air temperature <2.5 °C) rather than continuous explanatory variables. In this case, we used figures presented in McKenney and Pedlar (2003) to predict future forest productivity changes.

Productivity was also estimated using genecological models describing general species seed transfer and individual population growth response to climate (Aitken et al. 2008) that were developed using data from old provenance trials for white spruce, black spruce, jack pine, balsam fir, and white pine (Carter 1996, Cherry and Parker 2003, Andalo et al. 2005, Savva et al. 2007, Thomson and Parker 2008, Thomson et al. 2009). The models were derived from geographically relevant genetics studies that had provenances and/or planting sites from Ontario, Quebec, or the western U.S. Great Lakes States. Growth response functions for jack pine, white spruce, and black spruce populations from northeast Ontario were also available, enabling us to account for regional genetic differentiation among species in growth response to future climate. Details and sources of the genecological models used are presented in Table 2.

For all SI and genecological models (unless otherwise noted), current (1971–2000) and ensemble GCM projected climate values for Kapuskasing, Ontario were used as independent variables. Productivity response to climate change is expressed as the percentage change relative to the current estimated value for a given growth parameter (i.e., total height or basal area increment).

Results

Forest species composition

Bioclimate envelopes generated using climate projections of four GCMs generally indicated northward shifts in the core and marginal BCE over time under the A2 and B1 scenarios. Representative BCEs for black spruce for three future periods produced using these GCMs and the A2 scenario are presented in Figure 4. The relative geographic shifts in BCEs from current for a given period differed among GCMs, and were less extensive under the B1 than the A2 scenario. The largest difference (i.e., least spatial concurrence) between the current and future species BCE for a given period resulted from the MIROC32MR GCM, which forecast no core climate for this species in Ontario by the end of the century. For both the CGCM31 and MIROC32MR GCMs under the A2 scenario, over time the proportion of the core range within Ecodistrict 3E-1 decreased for boreal species but increased for GLSL species (Table 3). This resulted in an initial increase in total number of the 15 selected tree species (i.e., potential increase in species richness) with some portion of their core BCE within the ecodistrict. By 2100, however, the core BCE of most boreal species (though not necessarily the species themselves) occurred north of the ecodistrict, and the area was dominated by the core BCE of GLSL tree species.

Table 3. Percentage of the core bioclimate envelope of 15 boreal (a) and Great Lakes-St. Lawrence (GLSL) (b) forest tree species that occur in Ecoregion 3E-1 now and as projected for three future periods by the general circulation models CGCM31 and MIROC32MR under the A2 emission scenario.

Region/species	CGCM31				MIROC32MR			
	Current	2011-2040	2041-2070	2071-2100	Current	2011-2040	2041-2070	2071-2100
a. Boreal								
Balsam fir	100	100	67.7	0	100	1.9	0	0
Jack pine	100	97.1	64.9	0	100	9.9	0	0
Black spruce	100	93.9	0.2	0	100	0	0	0
White spruce	100	98.4	46.9	0	100	5.3	0	0
White cedar	96.8	100	99.8	0	97	65.6	3.3	0
Trembling aspen	100	100	100	0	100	100	46.2	0
White birch	95.7	100	100	13.5	95.3	100	100	0
b. GLSL								
Red pine	0	98.1	100	0	0	52.2	78.3	0
White pine	0.1	100	99.9	80.2	0	59.9	81.3	0
Sugar maple	0	0	99.0	74.1	0	0	73.8	15.5
Red maple	0	0	100	81.1	0	0	70.8	26.8
Black cherry	0	0	1.5	62.7	0	0	0	7.7
Yellow birch	0	48.3	97.2	0	0	8.5	67.8	0

Forest productivity: Site index models

Depending on the time period and SI model, predictions from ensemble climate projections indicated either increased or decreased height growth potential for boreal tree species. The SI models by Sharma et al. (in review) predicted major decreases in height for 40-year-old jack pine and black spruce plantations, with height growth potential decreasing over time, and this effect being more pronounced under the A2 scenario (Figure 5a). Under the same scenario, jack pine plantation height was predicted to increase slightly ($\approx 8\%$) during the first period, followed by a large ($>40\%$) decrease. Under the B1 scenario, jack pine height growth was predicted to stay the same or decrease slightly ($<20\%$) by 2100. Black spruce height growth decreased relatively more in all time periods under both scenarios. For both species, climate at the end of the century under the A2 scenario was predicted to be unsuitable for height growth (i.e., predicted height was negative). This extreme result may be partly explained by projected climate values that were beyond the range of plot sample climate data used to build the model.

Black spruce

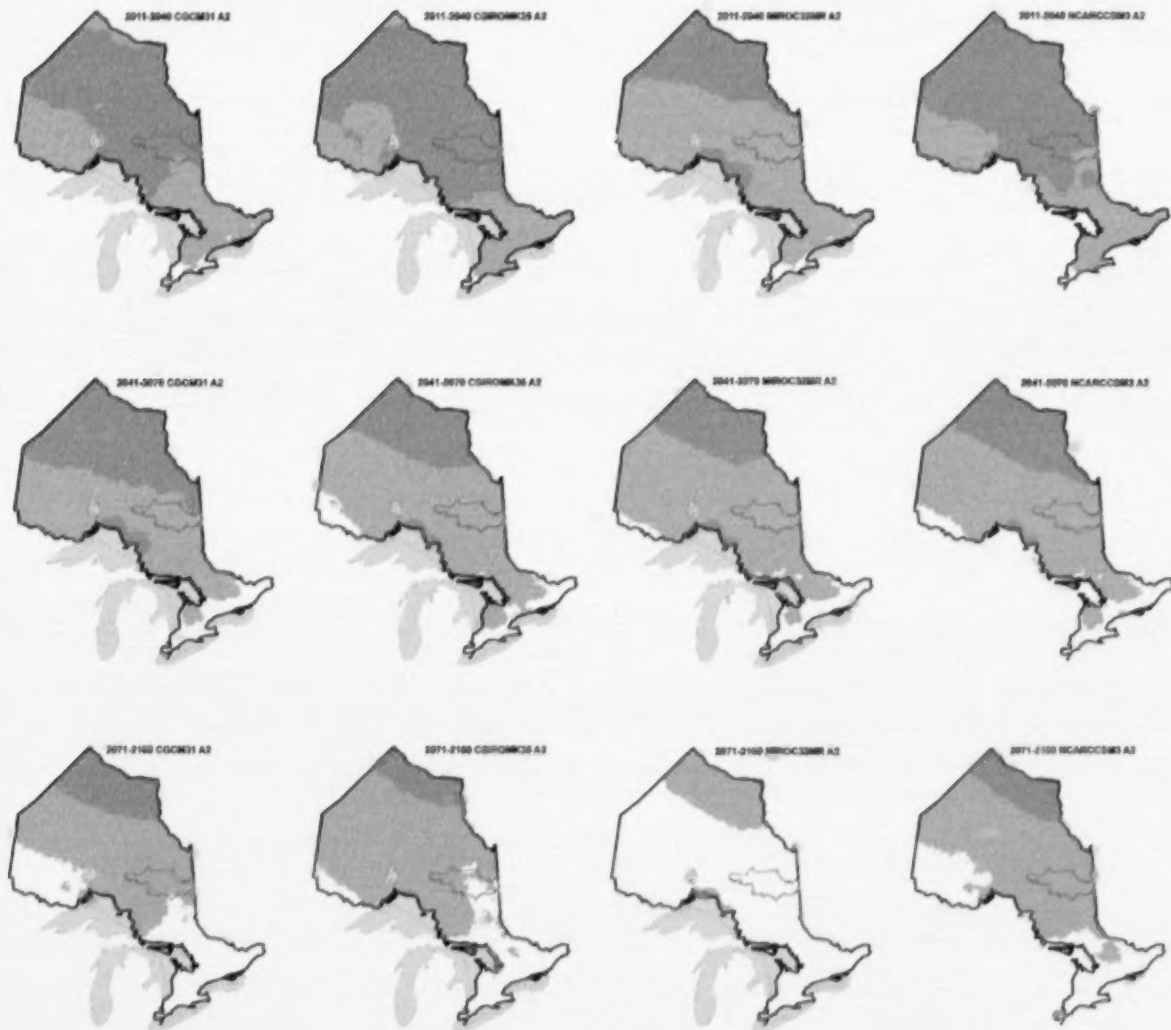


Figure 4. Bioclimate envelopes projected for black spruce by four general circulation models, the A2 scenario, and three future periods. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.

In contrast, the tree-based regression SI model for natural jack pine and black spruce stands (McKenney and Pedlar 2003) predicted modest to significant improvement in height growth of 50-year-old natural jack pine and black spruce stands under both the A2 and B1 scenarios (Figure 5b). For jack pine growing on deeper mineral soil (>40.5 cm deep), warmer future mean annual temperatures and wetter growing seasons were estimated to increase height growth by $\approx 9\%$ for all time periods and scenarios. Height growth response of black spruce to future climate varied with thickness of the organic layer. On upland sites with mineral soils and relatively thin (<24 cm) organic layers, height growth was estimated to increase up to 26% due to warmer, wetter growing seasons. Higher mean annual temperatures were predicted to improve height growth by 24 to 54% for black spruce on organic soils.

In comparison, biophysical SI models for natural stands of black spruce, balsam fir, trembling aspen, and white birch (Ung et al. 2001) predicted a general positive effect on future height growth, with the improvement in height becoming larger over time and under the A2 scenario (Figure 5c). Height of balsam fir, black spruce, and trembling aspen increased by 8 to 28%, depending on scenario and time period. By comparison, white birch showed a small initial decrease ($\approx 4\%$) but up to an 18% increase in height by 2100. We note that, similar to results of Sharma et al. (in review) for Ontario plantations, projected values for growing degree days and vapour pressure deficit were slightly beyond the range of empirical data used to derive the model, and predicted growth for 2071 to 2100 for all species should be interpreted with caution.

Forest productivity: Genecological models

Under future climate, general transfer functions predicted small ($<10\%$) to significant ($>20\%$) growth declines for white spruce, balsam fir, white pine, and jack pine (Figure 6). These functions estimate growth response using pooled data for a large number of populations (some from northern Ontario) planted in relatively few locations, and assume the shape and peak for individual populations do not vary significantly among planting environments (Carter et al. 1996, Andalo et al. 2005, Savva et al. 2007). Contrary to these results, population response functions for northeast Ontario populations of white and black spruce indicate modest to large increases in height growth with future warming (Figure 7a, c). This is likely because Ontario populations of these species currently inhabit areas with below optimum temperature for growth. Jack pine exhibits this same general geographic pattern in population response functions, but heights of northeastern Ontario populations were predicted to increase initially, followed by reduced heights under further increases in temperature (Figure 7b). That is, jack pine populations in northeastern Ontario are closer to equilibrium with current climate than are those of black or white spruce.

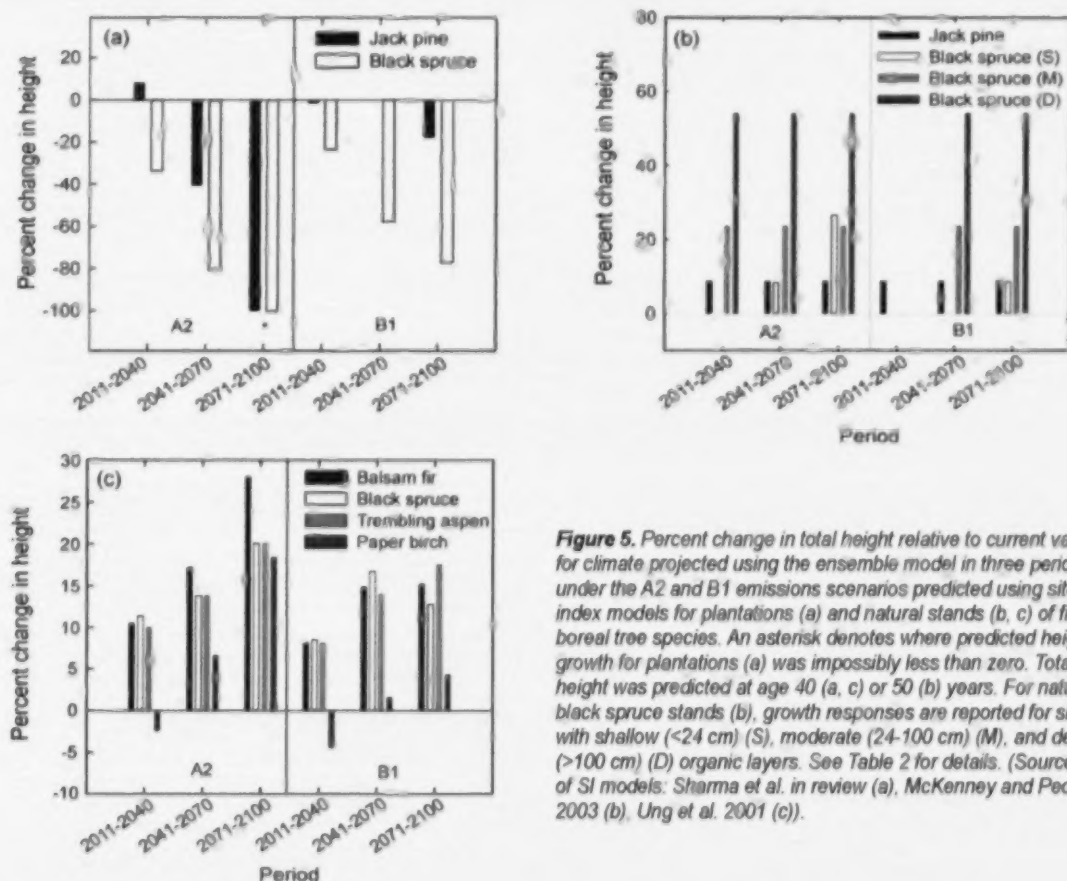


Figure 5. Percent change in total height relative to current values for climate projected using the ensemble model in three periods under the A2 and B1 emissions scenarios predicted using site index models for plantations (a) and natural stands (b, c) of five boreal tree species. An asterisk denotes where predicted height growth for plantations (a) was impossibly less than zero. Total height was predicted at age 40 (a, c) or 50 (b) years. For natural black spruce stands (b), growth responses are reported for sites with shallow (<24 cm) (S), moderate (24–100 cm) (M), and deep (>100 cm) (D) organic layers. See Table 2 for details. (Source of SI models: Sharma et al. in review (a), McKenney and Pedlar 2003 (b), Ung et al. 2001 (c)).

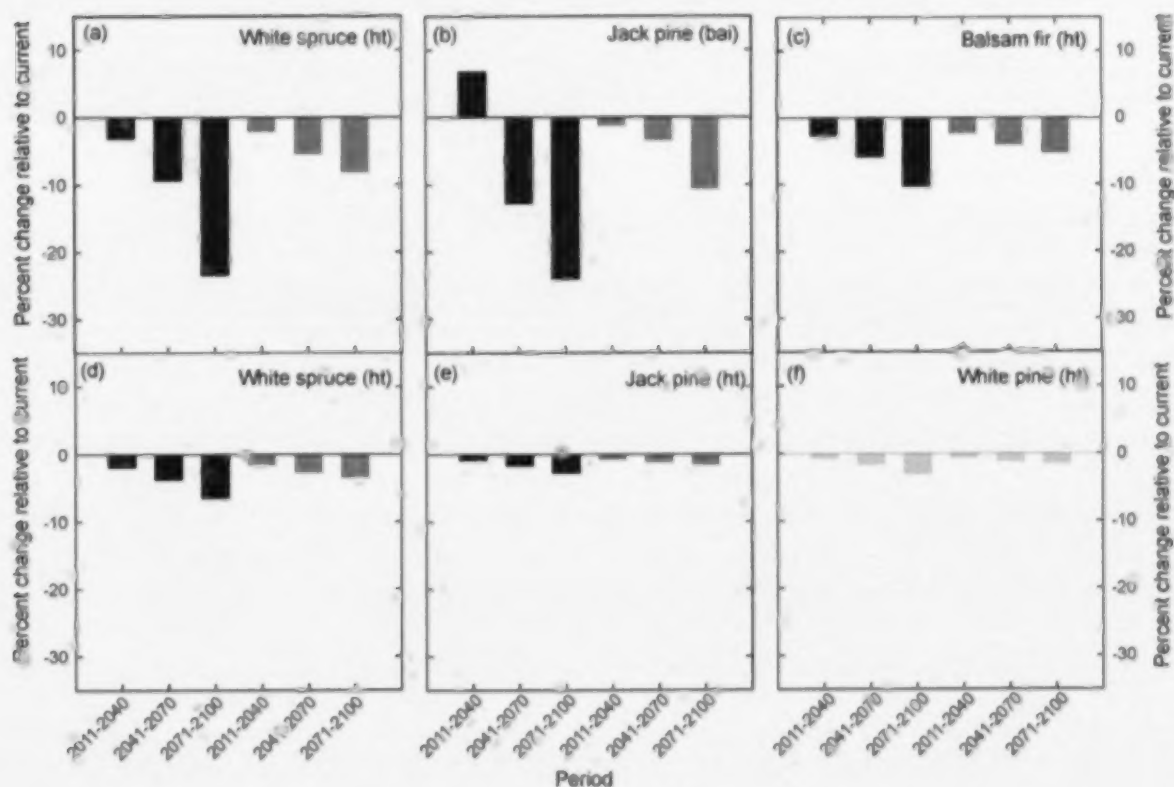


Figure 6. Percent change in total height (ht) or basal area increment (bai) relative to current values for climate projected using the ensemble model in three periods under the A2 and B1 emissions scenarios. Values are predicted using transfer functions from genecological models for white spruce (a, d), jack pine (b, e), balsam fir (c), and eastern white pine (f). Black bars represent the A2 and gray bars the B1 scenarios. See Table 2 for details. (Sources of transfer models: Andalo et al. 2005 (a), Savva et al. 2007 (b), Carter 1996 (c-f)).

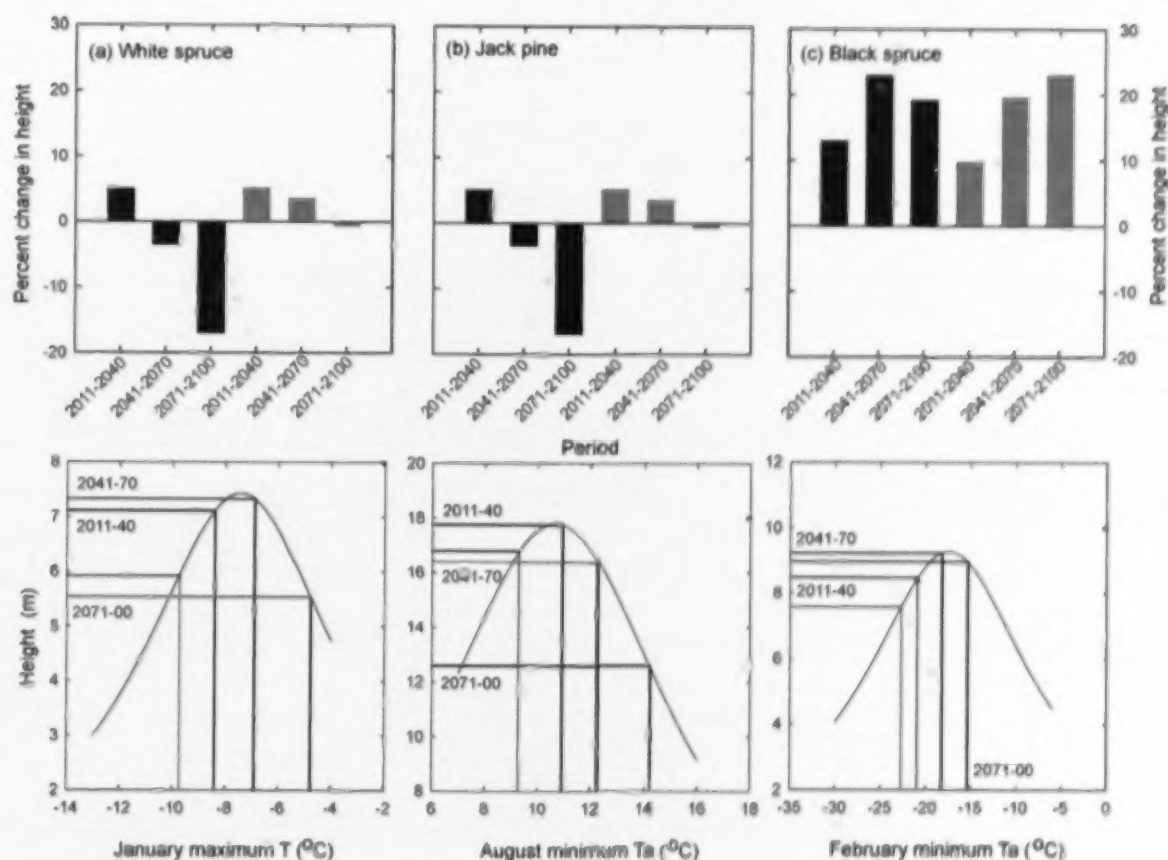


Figure 7. Percent change in total relative to current height for climate projected using the ensemble model for three periods under the A2 and B1 emissions scenarios. Values are predicted using population growth response functions for northeastern Ontario populations of white spruce (a), jack pine (b), and black spruce (c). Black bars represent the A2 and gray bars the B1 scenario. For each species, the corresponding line graph beneath the vertical bar graph presents the best-fit Cauchy function for the relationship of height with mean monthly maximum or minimum temperature (curved solid black line). In each graph, the solid red line denotes height for the current climate, while black lines identify predicted height for the future projected climate value for three periods under the A2 scenario. See Table 2 for details. (Sources of population response models: Cherry and Parker 2003 (a), Thomson and Parker 2008 (b), Thomson et al. 2009 (c)).

Discussion

Our vulnerability assessment of forest vegetation in Ecodistrict 3E-1 suggests that climate change will result in (1) altered forest composition due to individualistic, geographic shifts in the climatic habitat of major forest tree species and (2) small to moderate improvements in growth of several boreal tree species. However, predictions of climate change effects in this and other vulnerability assessments are subject to the inherent uncertainty associated with three primary factors. First, climate projections, especially when downscaled, vary with data quality and modelling assumptions of GCMs. This source of error may be reduced using ensemble GCM climate projections, rather than relying on those of a single GCM. Second, how effectively people will work to reduce future GHG emissions and how forest management can be used to affect adaptation of forest ecosystems are difficult to predict. Lastly, we have only a limited understanding of ecosystem processes and the complex manner of ecosystem response to changing climate; this can lead to overreliance on empirical observations and mechanistic model projections of climate change effects on forest ecosystems, and underestimation of the potential for acclimation, adaptation, and therefore, the stability of forest vegetation to climate change (Jump and Peñuelas 2005, Aitken et al. 2008). We considered these and other sources of uncertainty in assessing vulnerability and potential adaptation strategies for forest ecosystems of Ecodistrict 3E-1.

Forest species composition

A fundamental underlying assumption when predicting climate change effects on forest species composition is that species distribution is primarily controlled by climate (Woodward 1987). This assumption is supported by observations that recent climatic warming has resulted in species shifts to higher elevation (Beckage et al. 2008, Kelly and Goulden 2008) and northward (Lloyd et al. 2005, Payette 2007) for those with range restricted by low temperature. However, observed species distribution shifts in response to warming are sometimes counterintuitive (e.g., some species have moved to warmer conditions southward and downslope), and inconsistent with forecasted directional shifts in range with increased temperatures (Harsch et al. 2009, Lenoir et al. 2010, Zhu et al. 2012). These findings are only counterintuitive because of our current incomplete understanding of interactions among factors determining species distribution, underscoring the uncertainty inherent in our predictions of future forest composition (Lenoir et al. 2010, Angert et al. 2011, Dawson et al. 2011, Zhu et al. 2012).

Bioclimate niche modelling is a commonly used statistical approach that provides the best method for estimating broad scale range shifts due to climate change for many plant species (Iverson and Prasad 2002, Lawler et al. 2006). Niche-based models spatially delineate the climate space of plant species, or their fundamental climatic niche (McKenney et al. 2007). A weakness of this correlative approach is that it does not account for non-climatic factors (e.g., competition, herbivory, dispersal capacity) that act at finer spatial scales to define species distribution, i.e., the realized climatic niche (Pearson and Dawson 2003, Lawler et al. 2006, McKenney et al. 2007). Importantly, BCEs also do not account for genetic differentiation in climatic response among populations, nor do they account for the physiological effects of elevated CO₂ on biotic factors that influence species distribution (Loehle and LeBlanc 1996, Pearson and Dawson 2003, Mohan et al. 2007, Keenan et al. 2011). Although process-based species distribution models can account for many of these non-climatic factors, and have some predictive and statistical advantages, they have tremendously large data requirements that currently limit their application to relatively few species (Lawler et al. 2006, Morin and Thuiller 2009). Despite differences among approaches, species distribution projections from process-based and other models share commonalities with our results, with northward shifts in species climatic range, reduction or extirpation of suitable habitat for more northern species, and increased climatic suitability for more southern species as climate changes over time (e.g., Ehman et al. 2002, Xu et al. 2007, Iverson et al. 2008, Morin et al. 2008).

Given these concerns with niche-based species distribution models, the BCEs produced in our vulnerability assessment should be viewed only as a first approximation of the potential magnitude and broad pattern of the future climatic distributions of species within Ecodistrict 3E-1 (Pearson and Dawson 2003, Lawler et al. 2006). Since they depict projected shifts in a species' fundamental bioclimatic niche and do not account for dispersal capacity, they should not be viewed as maps of future species distribution. Migration rates of most tree species are expected to be too slow to allow them to remain synchronous with future climate, even where fragmentation of managed landscapes and natural physical barriers to migration are absent (Aitken et al. 2008). Therefore, the BCEs provided here do not project the future natural occurrence of GLSL species in Ecodistrict 3E-1.

Forest productivity

Forest productivity is governed primarily by temperature, precipitation, and radiation, as modified by stand (age, species, leaf area), site (nutrient availability, soil texture), and atmospheric CO₂ concentration. A recent review of satellite- and field-based studies examining changes in forest productivity associated with climate change since 1950 indicated that, at global scale, warmer temperatures and longer growing seasons have increased forest productivity when water is not limiting (Boisvenue and Running 2006). At regional and finer spatial scales, either positive or negative growth responses of northern forests to recent warming have been reported, depending on summer moisture availability (Lloyd and Fastie 2002, Wilmking et al. 2004, Beck et al. 2011). Recent satellite observations of some boreal regions of North America suggest drought stress and lower humidity may be contributing to decreased growth under warmer temperatures, a reversal of earlier (1982-1991) productivity gains (Myneni et al. 1997, Goetz et al. 2005, Lloyd and Bunn 2007, Beck and Goetz 2011). For example, radial growth response of black and white spruce to recent warming in Alaska was negative in warmer, drier interior regions, but positive in cooler, Coastal regions where water stress did not occur (Barber et al. 2000, Lloyd and Fastie 2002, Beck et al. 2011). Reduced growth with warming in higher density white spruce stands further supports the notion that temperature-induced drought may be a factor in these contrasting responses.

Results of other field studies show a strong interactive influence of temperature and moisture on tree growth and forest productivity. Since 1963, water deficits induced by climate warming have reduced productivity of western Canadian boreal forests through a combination of increased mortality and decreased growth of surviving trees (Ma et al. 2012). By comparison, in the eastern Canadian boreal forest, a combination of higher temperature and precipitation was associated with increased mortality, improved growth of surviving trees, and no net change in productivity over this same period (Ma et al. 2012). Wetter growing seasons generally increase stem radial growth of several boreal tree species in Ontario, with warmer spring temperature and increased summer moisture being particularly important (Hofgaard et al. 1999, Tardif et al. 2001, Hogg et al. 2008). Interannual variability in net ecosystem productivity (NEP) of boreal conifer and hardwood tree species also indicate warmer spring temperatures and longer growing seasons increase NEP where summer moisture availability is not limiting (Chen et al. 1999, Arain et al. 2002) and respiratory losses associated with warmer late summer conditions are modest (Arain et al. 2002, Piao et al. 2008, Wu et al. 2012). Collectively, these results suggest warmer temperatures will initially increase productivity of northern forests given sufficient moisture, but without increased moisture further warming may reduce productivity (D'Amigo et al. 2004, Wilmking et al. 2005, Aitken et al. 2008).

Changing climate may also increase tree mortality, thereby reducing stand and ecosystem productivity. Increased background mortality associated with recent changes in temperature and precipitation regimes have been reported for several forest regions (van Mantgem et al. 2009, Dietze and Moorcroft 2011, Peng et al. 2011). In Canada's boreal forests over the past 35 years, warmer temperatures and reduced moisture availability have increased mortality of jack pine, black and white spruce, aspen, and other species across a range of size classes, elevations, and latitude (Peng et al. 2011). Evidence of more frequent climate-driven episodic mortality (Allen et al. 2010) associated with extreme, seasonal, or multi-year drought and/or heat stress events is increasing, such as that documented for aspen in western Canada (Michaelian et al. 2010).

Atmospheric CO₂ enrichment will likely benefit forest productivity. As the substrate for photosynthesis, elevated CO₂ is expected to increase tree growth and improve water use efficiency, depending on species, site resources, and other factors (Hanson et al. 2005, Huang et al. 2007, Kubiske et al. 2007, Mohan et al. 2007, Ellsworth et al.

2012). Dendrochronological studies suggest that warm, moderately water-stressed ecosystems with an ample nitrogen supply may increase most with CO₂ fertilization, due to improved water use efficiency (Huang et al. 2007). Recent increases in CO₂ were associated in part with improved radial growth (Cole et al. 2010) and NEP of northern forests (Ollinger et al. 2002). In contrast, radial growth of several tree species in Ontario has reportedly declined over the past 50 to 60 years, despite improved water use efficiency due to elevated atmospheric CO₂ (Silva et al. 2010).

In our assessment, SI and genecological models were used to predict relative changes in growth in response to ensemble projected changes in climate for several tree species of Ecodistrict 3E-1 (Table 2). Two SI models predicted improved height growth under climate change, while a third estimated large reductions in height growth potential. However, all model projections are subject to error associated with unbalanced sampling intensity, which reduces the range of climate variables within the data set. This reduces their predictive power to estimate climate change effects for Ecodistrict 3E-1, and should be viewed as descriptive rather than predictive of future productivity of these tree species (Ung et al. 2009). As well, the regional models used here do not account for population differences in climatic sensitivity, which have been reported for aspen and other species (Monserud and Rehfeldt 1990, Chen et al. 2002, Anyomi et al. 2012).

Productivity of natural black spruce and jack pine stands is relatively low in northeastern Ontario due to cooler temperatures and, in the case of black spruce, its occurrence in areas with poor drainage and deep organic soils (McKenney and Pedlar 2003). In general, some warming will increase growth of trees currently occupying relatively colder (i.e., temperate and boreal species) regions, with deciduous species responding more than conifers (Way and Oren 2010). Site index models for several Ontario boreal species also indicated that growth will increase with warming (i.e., small to moderate increases in GDD), providing soil moisture is not limiting (Hamel et al. 2004, Beaulieu et al. 2011, Anyomi et al. 2012). Thus, climate warming will likely increase productivity of boreal tree species in the near- to mid-term so long as there is sufficient moisture (Rustad et al. 2001, Saxe et al. 2001, Wu et al. 2011). From these results and our model predictions, we conclude that slight warming will likely improve the growth of boreal tree species over at least the next several decades, but with further warming growth will likely decline (Wilmking et al. 2005, Aitken et al. 2008).

Genecological models of growth response to climate account for genetic differences in tree species that occur over a large geographic range. Their usefulness in a vulnerability assessment depends on the number of populations, or "seed sources" occurring in the area of relevance for which climate response models are available, and the range of climatic conditions over which the models have been tested. *General transfer functions* define the fundamental climatic niche of a species based on the pooled growth response of multiple seed sources to a range of climatic conditions by planting them on a number of test sites. General transfer functions used in this assessment generally predicted slight to moderate growth reductions under a future climate for white spruce, jack pine, balsam fir, and white pine. By comparison, *population growth response functions* define the fundamental climatic niche of individual seed sources and therefore provide information not available from general transfer functions, i.e., they quantify population- rather than species-level growth responses to climate (Aitken et al. 2008). Population response functions for northeastern Ontario populations of black and white spruce predicted slight to moderate increases in height growth under future climate conditions in Ecodistrict 3E-1. By comparison, height growth of jack pine was predicted to initially increase slightly with warming, and then decrease as the warming continued. This is similar to growth responses predicted for jack pine by Savva et al. (2007). Collectively, these population response functions predicted that height growth of Clay Belt populations of these three species will benefit from warmer temperatures over the next several decades. Increased growth of jack pine, however, will likely be relatively short-lived.

Tree species populations inhabiting environments cooler than their thermal optimum have been demonstrated for several species beyond those included in our assessment (Carter 1996, Rehfeldt et al. 1999, Reich and Oleksyn 2008). This may occur because of lags in either adaptation or migration, where these populations have not kept pace with the rate of post-glacial climatic change (Namkoong 1969, Aitken et al. 2008). Alternatively, under the harsh environmental conditions occurring at the northern portion of these species' range, natural selection for survival or reproduction may be favoured over vigorous height growth. Our modelling results suggest

that for the boreal species examined here, where current temperature is below the species or population optimum for growth, a slightly warmer climate will increase tree growth until this thermal optimum is exceeded, beyond which growth will decline with further warming. These temperature-induced growth declines will develop more quickly and be proportionally larger where precipitation does not change or decreases. As CO₂ concentration increases, its beneficial effects on water use efficiency and the temperature optimum for photosynthesis may moderate some of the adverse climatic influences on growth.

Adaptation approaches

Rural, forest-based communities of the Clay Belt that rely on climate sensitive northern forest ecosystems are vulnerable to changes in climate (Davidson et al. 2003, Lynn et al. 2011). Climate change threatens these communities through a variety of effects that reduce forest productivity and wood supply, such as increased disturbance by wildfire, insects, and extreme weather. In particular, the unique rights, cultures, and economies of First Nations are intimately connected to the land and are particularly vulnerable (Chapin et al. 2004, Peach Brown 2009, Lynn et al. 2011). Evidence that effects of climate change are already occurring (Parmesan 2006, IPCC 2007) underscores the importance of planning to enhance forest adaptation. Conducting vulnerability assessments, incorporating climate change into natural resource management planning, expanding monitoring, and developing adaptation strategies are necessary steps in this process.

Since silviculture is applied to <1% of the commercial forest landbase of Ontario annually², most forests will need to adapt autonomously to climate change, with only a relatively small area available for management intervention to influence the direction and location of adaptation efforts. Given the uncertainty of both climate change and ecosystem response to the changes, no single approach will apply in all situations. Rather, a toolbox approach to adaptation will be needed, whereby various short- and longer-term options can be selected and combined to create individualized adaptation strategies tailored to specific situations (Spittlehouse and Stewart 2003, Millar et al. 2007, Williamson et al. 2009). Several forest adaptation options have been identified that can be broadly classified as deterministic and indeterministic (Millar et al. 2007). Deterministic approaches assume future climate is acceptably predictable and consist of forest practices that enable forests to temporarily resist or recover from expected environmental change. Alternatively, indeterministic adaptation approaches assume future climate is uncertain and cannot be suitably predicted, and activities are aimed at promoting the natural adaptive processes of ecosystems that enable them to accommodate and respond to change (Millar et al. 2007). A thorough overview and consideration of forest adaptation options that can be applied at the strategic and operational level were offered by Ogden and Innes (2007). They incorporated climate change adaptation into forest management planning using the criteria for conservation and sustainable management of temperate and boreal forests as defined by the Montreal Process.

Based on our assessment of the vulnerability of Clay Belt forests to changes in species composition and productivity, we offer several recommendations for adaptation. We confine our recommendations to a subset of adaptation strategies directly relevant to effects on forest species composition and productivity as projected in our study.

Assisted migration to improve forest regeneration and productivity

For many people, an obvious suggestion for adapting forests to climate change is northward relocation of seed sources or even introduction of non-local species adapted to warmer temperatures (Ste-Marie et al. 2011). However, implementing this strategy may require considerable investment in infrastructure, since non-local seed may not be easily obtained (Pedlar et al. 2011). Consideration of the size of the commitment needed to meaningfully affect forests is also needed, since each year the area of forest planted is relatively small compared to the total forested area. Most importantly though, before undertaking assisted migration practitioners should weigh the potential risks against possible benefits (Aubin et al. 2011).

² Based on average total area of Crown land harvested, planted, and direct seeded from 2001 to 2010 within the 43.8 million ha Area of Undertaking. Source: National Forestry Database, <http://nfdp.ccfm.org/index>

Assisted migration is most likely to be beneficial if it is done with knowledge of the genetic adaptation of seed sources to climate. This requires what is referred to as genetic climate niche modelling (GCNM), which combines modelling the bioclimatic niche for individual seed sources of a species with detailed knowledge of the optimal climate for those seed sources. This is a powerful tool for evaluating how populations (seed sources) respond to climate that allows them to be planted where the risk of climatic stress is reduced. Before considering an operational assisted migration program, GCNM should be undertaken, with the following steps:

- 1) Assess risk from climate change to individual seed sources based on current status of the species (e.g., rarity, age structure, ecological factors affecting natural and assisted regeneration, extent of response to possible warmer/drier future climate).
- 2) For seed sources of concern identified in Step 1, evaluate the future range of possible climates using GCM(s) and GHG scenario(s).
- 3) Where the threat to a species or populations is sufficient to warrant considering assisted migration, develop options for obtaining sufficient seed to implement it.

Assisted migration is not recommended unless the risks to local seed sources are understood and a plan is developed to obtain seed from geographic sources adapted to future climate conditions. Since uncertainty about the rate of climate change is considerable and tree species in Ontario have life spans well beyond 100 years, seed source selection will inevitably involve tradeoffs between adaptation to near-term conditions and adaptation to future climates (O'Neill et al. 2008).

Managing for ecological resilience

Are there actions that might adapt forests in the short- to medium term to the effects expected from rapid, unexpected ecosystem changes in response to an uncertain and variable future environment? Some forest ecologists suggest that increasing the ecological resilience of forests may make them less vulnerable and better able to cope with stresses from climate change. Managing forest ecosystems to maintain or restore ecological resilience may provide a precautionary approach to forest management to allow these ecosystems to respond and better autonomously adapt to future climate (Chapin et al. 2004, Puettmann 2011). This indeterministic approach to adaptation accepts that the speed, severity, and effects of climate change are unpredictable, and emphasizes the need to manage for ecological resilience to facilitate the natural capacity of forest ecosystems to cope with future environmental change (Millar et al. 2007, Campbell et al. 2009, Puettman 2011).

Forest management has traditionally been guided by the concepts of ecosystem stability, linear successional change, and sustained yield of selected products, based on the assumption that forest ecosystems are predictable and controllable. From this standpoint, adaptation consists of silvicultural practices aimed at reducing vulnerabilities to specific perceived risks and deriving narrowly defined, short-term human benefits, such as increasing vigour and maintaining productivity of commercial species in a drier climate through more intensive silviculture. Under a resilience framework for adaptation, ecosystems are viewed as complex adaptive systems that may organize around several possible alternative stable states, in response to inevitable environmental change. An approach that involves managing for complexity and ecological resilience accepts that ecosystems are constantly changing and seeks instead to improve the natural capacity of forests to respond and adapt to a wide variety of environmental conditions (Chapin et al. 2004, Puettmann 2011). In theory, enhancing ecological resilience allows ecosystems to absorb disturbance and reorganize while retaining essentially the same function, structure, identity, and feedbacks. This approach allows for short- to mid-term adaptive strategies centred on avoiding thresholds for shifts to alternative states, and maintaining the current system state. However, in the longer term when thresholds for state change are exceeded, adaptation efforts can direct ecosystem reorganization and transformation to an alternative state (e.g., shift from boreal conifer to GLSL conifer in Ecodistrict 3E-1) better suited to a new climate. Widespread decline of aspen associated with recent warming in some boreal forest regions of western Canada may be an expression of the early stages of a climate change-induced transformation of these ecosystems to an alternative stable state (Hogg et al. 2008).

Resilience is derived from the inherent resource capital of an ecosystem, the heterogeneity of ecosystems on the landscape, and the species, functional, and genetic diversity of the ecosystem (Chapin et al. 2004). In cases of biodiversity loss through intensive forest management or human-generated stressors (e.g., pollution, invasive species), the complex web of interactions between organisms may be disrupted, potentially inhibiting adaptation to climate change (Mooney et al. 2009). Certainly, the structure and age-class distribution of natural, pre-industrial forest landscape of Ecodistrict 3E-1 has undergone significant change over the past century (Carleton 2000, Pinto et al. 2008, Cyr et al. 2009), perhaps resulting in reduced resilience (Noss 2001, Drever et al. 2006, Cyr et al. 2009, Puettmann 2011). The advantage of focusing forest management on maintaining complexity, biodiversity, and ecosystem processes rather than stable delivery of specific forest products is viewed as analogous to an insurance policy. The insurance hypothesis presumes that biodiversity insures ecosystems against loss of function under environmental fluctuations because higher numbers of species increase the probability that some will remain productive even if others fail (Yachi and Loreau 1999). In this way, ecological redundancy may buffer year-to-year changes in productivity due to climate variability, enhancing productivity over the longer-term (Yachi and Loreau 1999). "Payment" for this insurance policy comes in the form of immediately reduced average harvest levels to give a better chance at higher long-term average productivity, along with continued ecological function of forests (Puettmann 2011).

An adaptation strategy based on increasing forest resilience relies in part on emulating natural disturbance to mimic the range of natural biodiversity, using a portfolio of silvicultural approaches proposed to foster resilience (Lindemayer and Franklin 2002, Drever et al. 2006). Where appropriate, this can mean reduced reliance on conventional even-aged management and increased use of structural and variable retention partial harvest systems (Lindemayer and Franklin 2002, Puettmann 2011). Other actions to maintain or increase resilience include protecting an adequate (natural) amount of old-growth forests, avoiding fragmentation to maintain connectivity, minimizing conversion of natural forests to intensively managed plantations, emulating natural disturbance patterns at stand and landscape scales, and maintaining diverse gene pools (Noss 2001, Lindemayer and Franklin 2002, Millar et al. 2007, Campbell et al. 2009). While further research and testing are needed to support more widespread use of natural disturbance emulation in general and resilience in particular (Perera et al. 2004, Drever et al. 2006, Puettmann et al. 2009, Kuuluvainen and Grenfell 2012), we consider managing for resilience to be a no-regrets strategy for climate change adaptation that will benefit forest ecological sustainability despite uncertainty about the speed of climate change and specific effects on forests.

Our analysis of potential adaptation options to reduce climate change effects on forests for the most part does not address changes to infrastructure (e.g., fire suppression capacity, road and water crossing engineering) that should be considered (Peach Brown 2009, Johnston and Hessel 2012). We also do not address needs for social adaptation, though this is also an area of importance, especially for Ontario's geographically isolated First Nations communities located in the northern part of the Boreal forest region.

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Appendix 1.

Representative current and future bioclimate envelope maps for six forest tree species of Ontario generated using the CGCM31 and the A2 scenario.

Trembling aspen

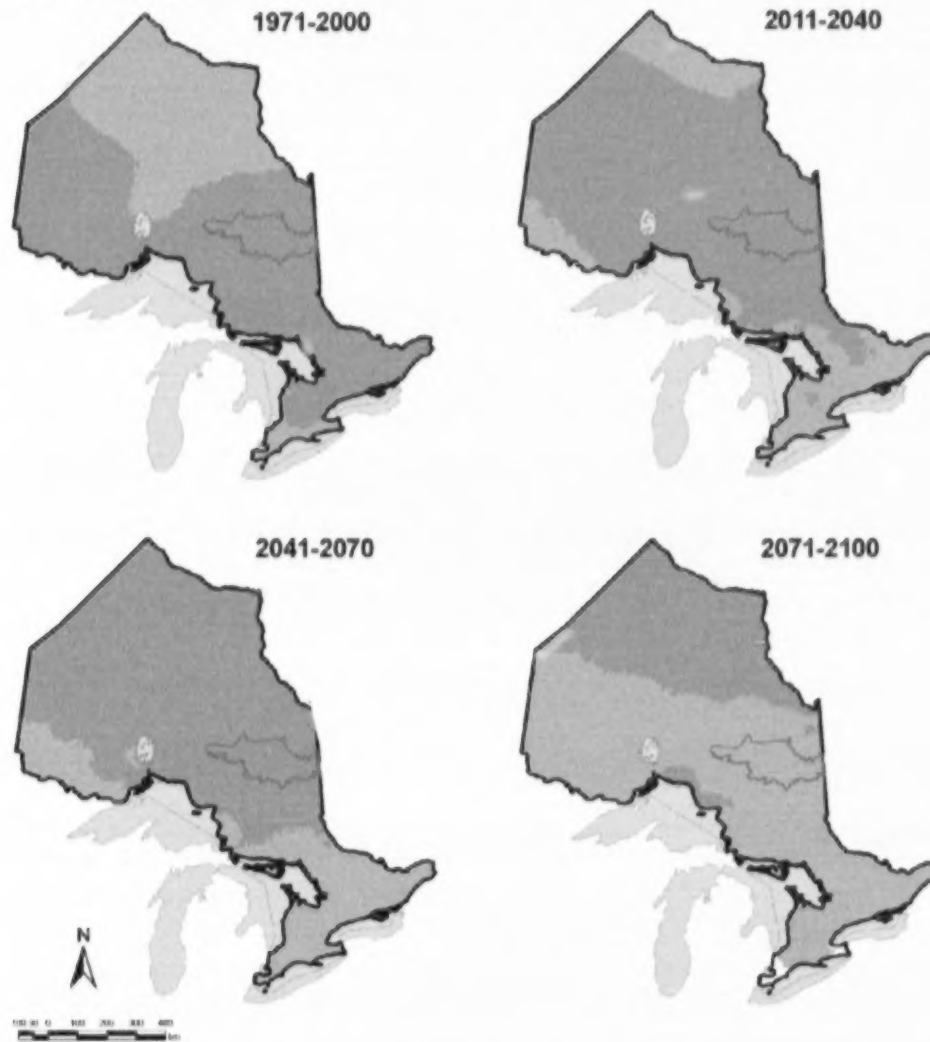


Figure A1. Current and future bioclimate envelopes for trembling aspen in Ontario and Ecodistrict 3-1 projected using CGCM31 and the A2 scenario. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.

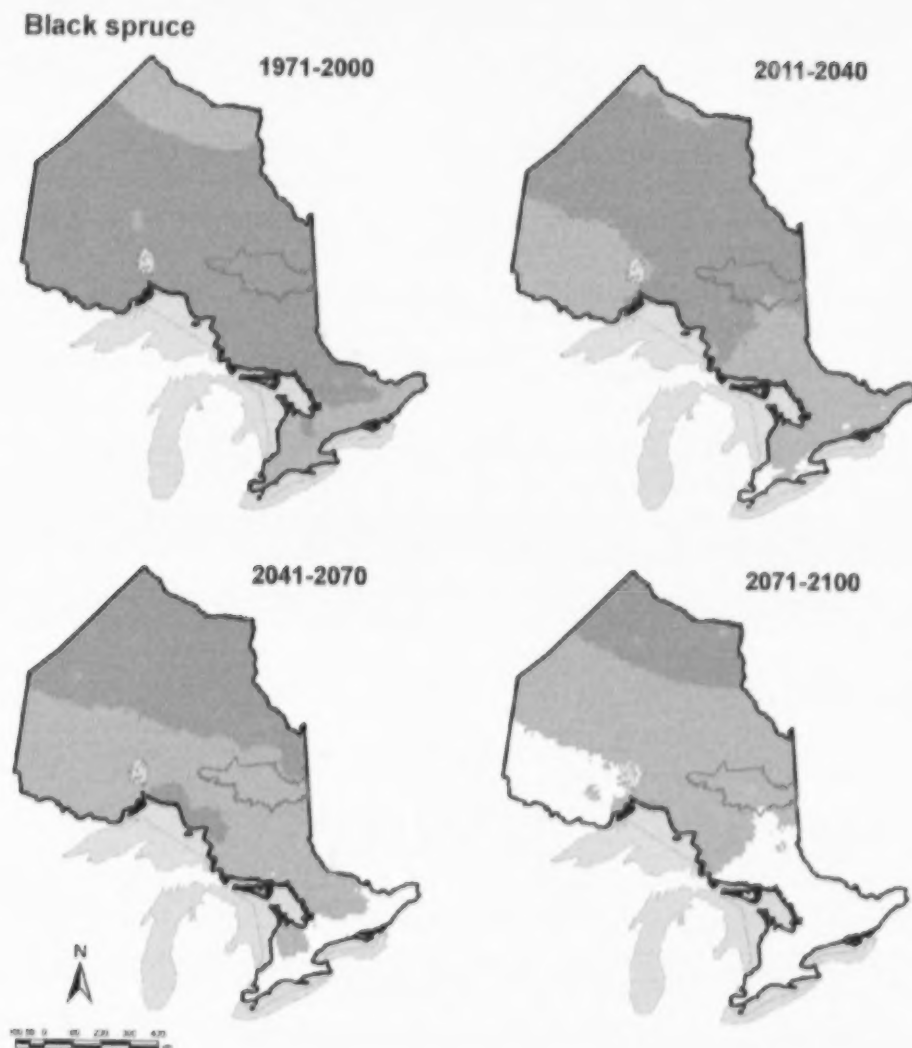


Figure A2. Current and future bioclimate envelopes for black spruce in Ontario and Ecodistrict 3-1 projected using CGCM31 and the A2 scenario. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.

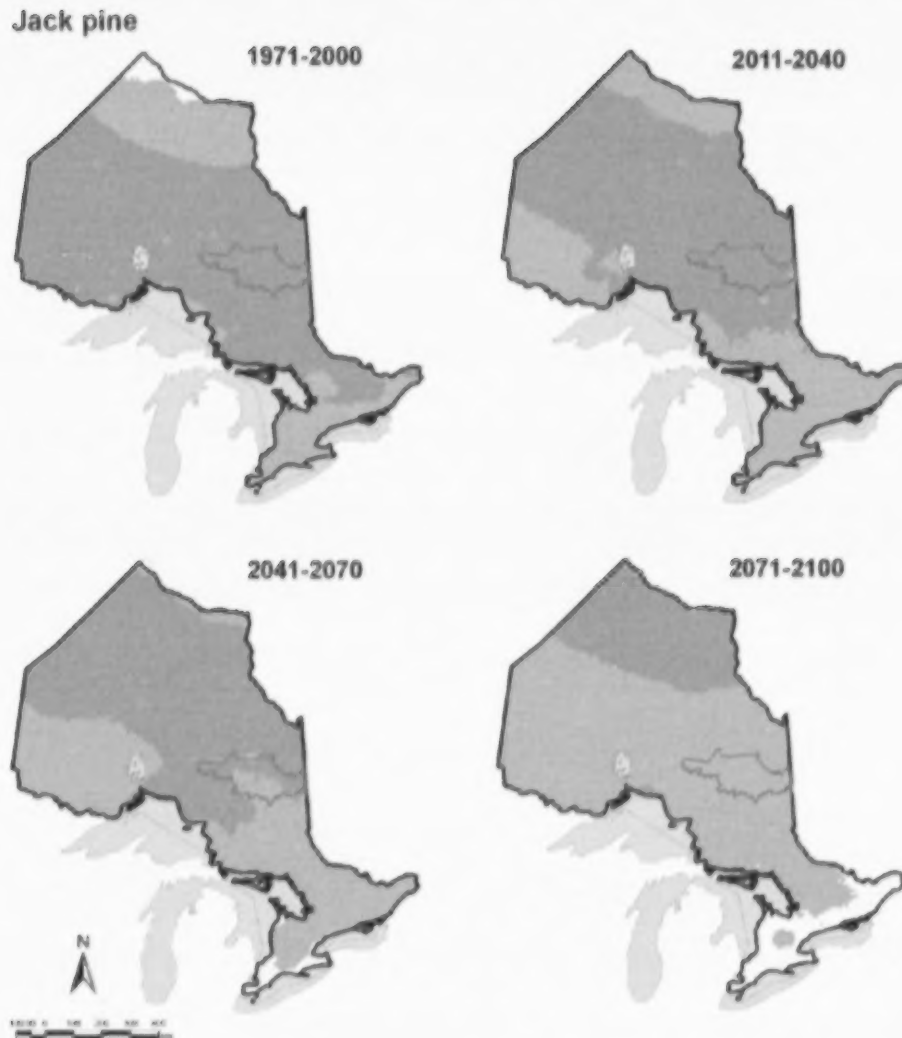


Figure A3. Current and future bioclimate envelopes for jack pine in Ontario and Ecodistrict 3-1 projected using CGCM31 and the A2 scenario. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.

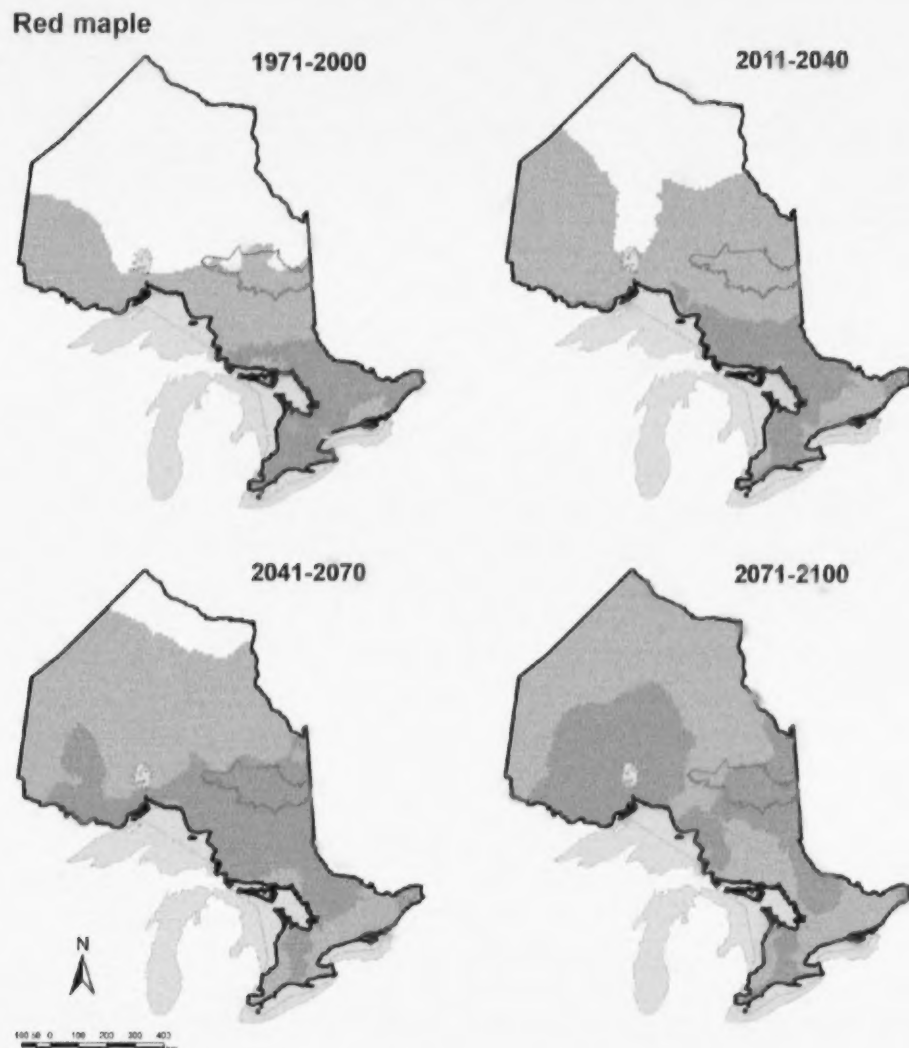


Figure A4. Current and future bioclimate envelopes for red maple in Ontario and Ecodistrict 3-1 projected using CGCM31 and the A2 scenario. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.

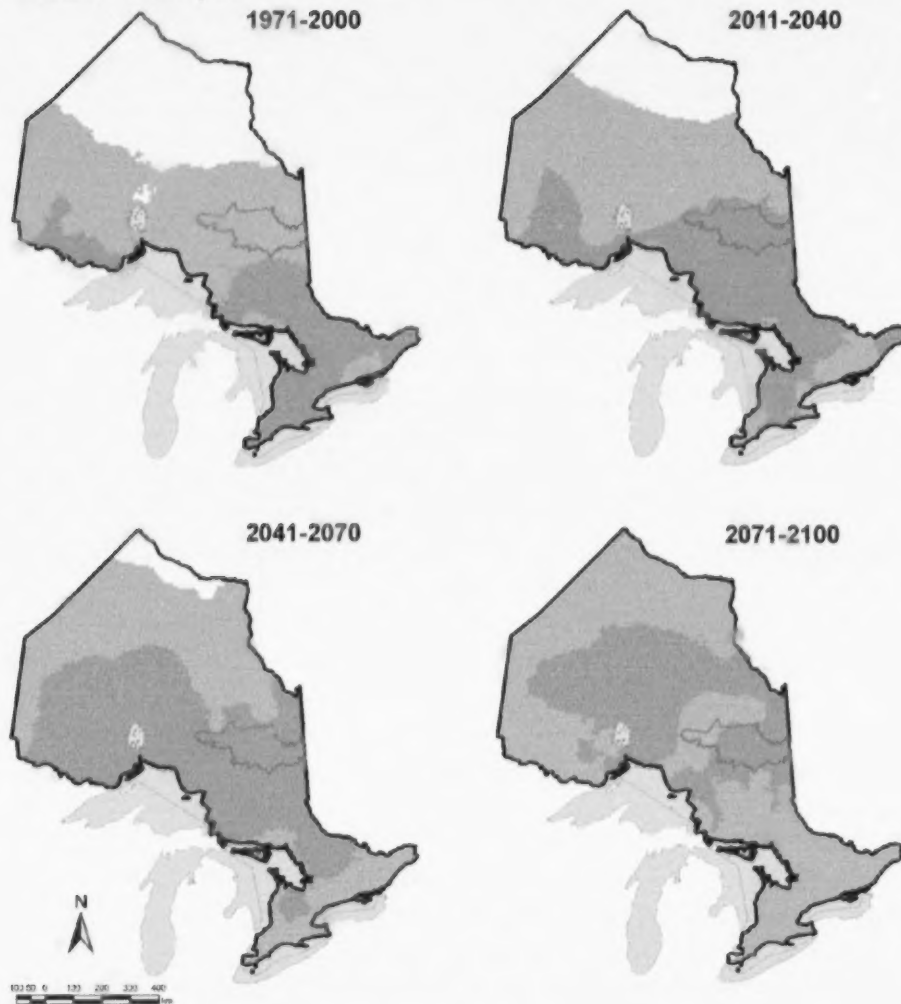
Eastern white pine

Figure A5. Current and future bioclimate envelopes for eastern white pine in Ontario and Ecodistrict 3-1 projected using CGCM31 and the A2 scenario. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.

Yellow birch

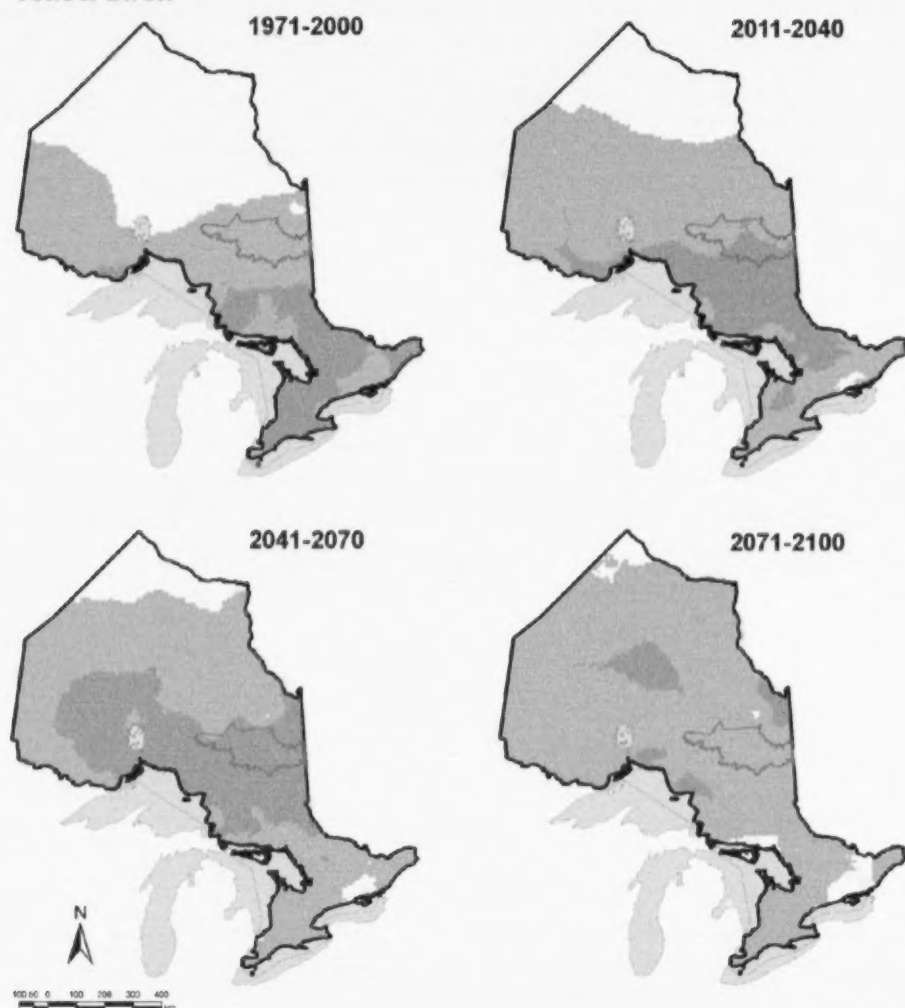


Figure A6. Current and future bioclimate envelopes for yellow birch in Ontario and Ecodistrict 3-1 projected using CGCM31 and the A2 scenario. Green represents core climatic range, brown denotes marginal climatic range, and white indicates where the species does not occur. The red line encloses the area of Ecodistrict 3E-1.



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